The ecological effects of urbanization of coastal watersheds: historical increases in nitrogen loads and eutrophication of Waquoit Bay estuaries

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Abstract: Historical changes in land use on coastal watersheds have increased rates of land-derived nitrogen loading to estuaries and altered their biogeochemistry and food webs. We used information on human populations and land uses within the watershed of Waquoit Bay, Cape Cod, Massachusetts, U.S.A., to model how nitrogen loads derived from atmospheric deposition, fertilizer use, and wastewater disposal have changed since the 1930s. Nitrogen loading into Waquoit Bay more than doubled between 1938 and 1990. The predominant source of nitrogen added to the bay changed from atmospheric deposition to wastewater disposal during the 1980s, reflecting the increasing urbanization of Cape Cod. Larger nitrogen loads increased nitrogen concentrations in the water, altering the assemblage of primary producers and resulting in eutrophication of the estuary. Biomass of phytoplankton and macroalgae increased, and areal cover of eelgrass (*Zostera marina*) decreased, with increasing nitrogen load. An increase in nitrogen load from 15 to 30 kg $N \cdot ha^{-1} \cdot year^{-1}$ virtually eliminated eelgrass meadows. Land-use changes prompted by urban sprawl can therefore be linked to marked changes in water quality and eutrophication of receiving waters.

Résumé : Les changements historiques d'utilisation des terres dans les bassins versants côtiers ont augmenté les taux de charge de l'azote d'origine terrestre dans les estuaires et y ont modifié la biogéochimie et les réseaux alimentaires. Des données sur les populations humaines et l'utilisation des terres dans le bassin versant de la baie de Waquoit du cap Cod au Massachussetts, États-Unis, ont permis de modéliser comment les charges d'azote reçues par précipitation atmosphérique, les engrais et les eaux de rejet ont changé depuis les années 1930. La charge d'azote à la baie de Waquoit a plus que doublé entre 1938 et 1990. Durant les années 1980, la source principale d'azote dans la baie est passée de la précipitation atmosphérique d'azote à l'azote des eaux de rejet, un reflet de l'urbanisation croissante du cap Cod. Les charges accrues d'azote ont augmenté les concentrations d'azote dans l'eau, ce qui a modifié la composition des producteurs primaires et résulté en l'eutrophisation de l'estuaire. Les biomasses du phytoplancton et des macroalgues ont augmenté avec l'apport plus important d'azote, alors que la surface couverte par la zostère (*Zostera marina*) a diminué. Les charges accrues d'azote de 15–30 kg $N \cdot ha^{-1} \cdot année^{-1}$ ont presque complètement éliminé les marais à zostères. Les changements dans l'utilisation des terres consécutifs à l'étalement urbain peuvent donc être reliés à des modifications marquées de la qualité de l'eau et à l'eutrophisation des eaux réceptrices.

[Traduit par la Rédaction]

Introduction

Enrichment of coastal waters from land-derived nitrogen is one of the most pervasive threats to aquatic ecosystems today (Carpenter et al. 1998). Human-induced alterations to coastal watersheds have invariably resulted in changes in the ecology of aquatic systems. Nitrogen typically limits the growth of estuarine producers (Ryther and Dunstan 1971; Howarth 1988) and increases in the supply of nitrogen induce eutrophication in coastal waters (Nixon 1995).

In a large geographical context, all available nitrogen is derived from atmospheric sources through N_2 fixation. Nitrogen in fertilizers is fixed from N_2 gas and is then applied to major crops. The food we ingest transforms the nitrogen derived from fertilizer into nitrogen that is released as wastewater. On a watershed scale it is helpful to separate nitrogen delivery into three components: atmospheric deposition both within and outside the watershed, fertilizer application on lawns, turf, and agriculture, and disposal in human-derived wastewater (Lee and Olsen 1985; Cole et al. 1993). Some portion of the nitrogen that is deposited on watersheds from these sources is then delivered from land to coastal systems (Hinga et al. 1991; Correll et al. 1992).

Numerous mechanistic and empirical models have been developed to predict both the amount of land-derived nitrogen entering surface water and groundwater and the ecological effects of that nitrogen on the receiving water body. These range from relatively simple models based on human population densities within watersheds (Cole et al. 1993) to more complex models that include terms for atmospheric inputs and land-use classifications (Valiela et al. 1997*a*). We used one such model to reconstruct changes in nitrogen loading to the estuarine complex in Waquoit Bay, Cape Cod, Massachusetts, U.S.A. (Fig. 1).

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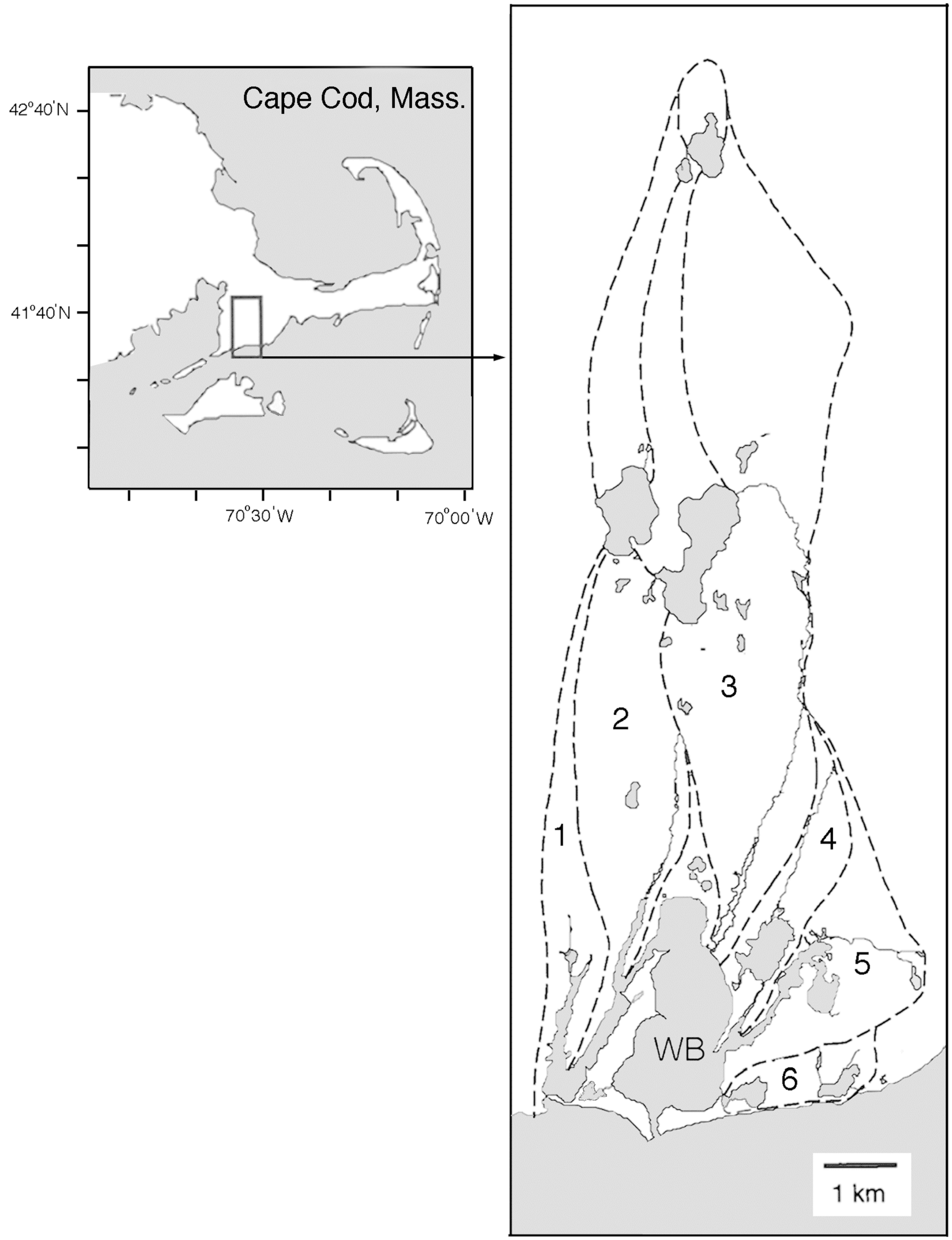
Fig. 1. Location of Waquoit Bay (WB), Cape Cod, Massachusetts, U.S.A., and its subwatersheds (1–6). Subwatersheds were delineated using a MODFLOW groundwater transport model by J. Brawley and C.-H. Sham, and are reported along with measured nitrogen loads in Valiela et al. (2000*a*). Nitrogen loads for each subwatershed are as follows: (1) Eel River, 93 kg·ha^{-1}·year^{-1}; (2) Childs River, 601 kg·ha^{-1}·year^{-1}; (3) Quashnet River, 403 kg·ha^{-1}·year^{-1}; (4) Hamblin Pond, 15 kg·ha^{-1}·year^{-1}; (5) Jehu Pond, 16 kg·ha^{-1}·year^{-1}; (6) Sage Lot Pond, 12 kg·ha^{-1}·year^{-1}.



The pattern of changing land use in the Waquoit Bay watershed is representative of regional trends. Agricultural land in the northeastern U.S.A. has decreased from 4.6 million ha of major crops in New England and New York in 1910 to only 1.5 million ha in 1995. In the same region livestock decreased from 3.2 million head to just over 2 million head during the same time period (U.S. Department of Agriculture (USDA) 1993, 1999).

Despite the decrease in agricultural area, the pressing need to increase crop yields and the rise in industrial manufacturing of synthetic fertilizers since the 1940s have fostered an increase in agriculturally derived nitrogen loading to most U.S. waters. In recent decades, fertilizer has accounted for up to 60% of net nitrogen inputs to land in the U.S.A. (Jordan and Weller 1996). Nitrogen from fertilizers accounts for only 19% of the total nitrogen inputs to the northeastern U.S.A., but the net import of food and livestock feed accounts for another 32% of nitrogen inputs to the region (Jordan and Weller 1996). This importation, another consequence of the decrease in agricultural land, converts nitrogen from synthetic fertilizers that are produced outside the region to nitrogen in wastewater that is deposited locally. Since 99% of ingested nitrogen is excreted (Smil 1999), nearly 420 million kg N·year^{-1} is imported into the watersheds of the northeastern U.S.A.

There has been reforestation of some of the agricultural land in the New England area (Foster 1992), but much of the land has given way to urban expansion, particularly in coastal regions. The populations of coastal counties in the U.S.A. are increasing three times faster than the U.S. population as a whole (Culliton et al. 1989). Over the last 60 years the population of Cape Cod has exceeded even that rate, increasing by a factor of nearly 5, from 32 000 in 1930 to over 180 000 in 1990 (www.census.gov). As a result of the changes in land use and agricultural practices, substantial changes have occurred in the amount of nitrogen from human NO_x emissions (Holland et al. 1999), fertilizers (Smil 1997), and wastewater both in the northeastern U.S.A. and on Cape Cod. Changes in nitrogen loading and the resultant alterations in estuarine ecology in Waquoit Bay might mirror, as a microcosm, changes that have occurred throughout the northeastern U.S.A.

In this paper we use land-cover data and reconstructed rates of atmospheric deposition and fertilizer use to develop regional changes in the quantity of nitrogen from the three major sources to Waquoit Bay during the last two-thirds of the 20th century. We also demonstrate that the changes in nitrogen loads from the Waquoit Bay watershed were sufficient to alter the ecology of the estuaries by assessing the effects of urbanization on ecological indicators, including phytoplankton, macroalgae, seagrasses, and scallops. The reconstruction that we present in this paper defines, within the limits of available data, how the nitrogen load into Waquoit Bay has changed since 1938. Long-term monitoring of nutrient inputs to most coastal systems is rare, and historical records of primary production are difficult to find. The reconstruction of land use and resultant nitrogen loads into Waquoit Bay offer the opportunity to understand that temporal setting.

Methods

To recreate historical nitrogen loads we needed a model that was complex enough to incorporate changes in each source of nitrogen, but that was not so complex that it required arcane data which would be impossible to recreate from scant historical records. We applied a previously developed (Valiela et al. 1997*a*) and field-verified (Valiela et al. 2000*a*) nitrogen-loading model (NLM) to reconstruct the changes in nitrogen loading to Waquoit Bay since the 1930s.

The Waquoit Bay NLM (Fig. 2) predicts nitrogen inputs to watersheds from atmospheric deposition, fertilizer use, and wastewater disposal, and then calculates the groundwater-delivered nitrogen loads into estuaries (Valiela et al. 1997*a*, 2000*a*). NLM was derived from literature reviews and research in the Waquoit Bay estuarine system. It includes terms for each of the major sources of nitrogen and applies each of these sources to the different land uses in the watershed, tracking the losses and transformations that occur as the nitrogen moves through the landscape (Fig. 2). One example of how NLM works is that 65% of nitrogen from atmospheric sources deposited on forests is lost as it travels through forest biomass and soils, 61% of the remaining nitrogen is lost in the vadose zone, and then 35% of the surviving nitrogen is lost in the aquifer before the remainder reaches the Bay at the seepage face (Valiela et al. 1997*a*, 2000*a*).

We have compared the structure and performance of NLM and six other published loading models (I. Valiela, J.L. Bowen, and K.D. Kroeger, unpublished data). In terms of model structure, NLM is among the most inclusive of transformation and transport terms. Its structure (Fig. 1) mirrors the path and fate of nitrogen moving through watersheds in a biogeochemically comprehensive fashion. In terms of performance, the predictions of nitrogen loads from watersheds made by NLM are statistically indistinguishable from measured nitrogen loads (Valiela et al. 2000*a*); the other models made predictions that required post-hoc correction terms of 26–76% to accurately match measured values (I. Valiela, J.L. Bowen, and K.D. Kroeger, unpublished data).

Using NLM to calculate historical nitrogen loads required that we compile data on changes in the land-use mosaic of the Waquoit Bay watershed, as well as on historical trends in atmospheric deposition, fertilizer use, and wastewater disposal, the three major sources of nitrogen. Below we address how each of these was estimated.

Changes in the land-use mosaic

To model nitrogen loads into receiving estuaries, NLM requires inputs of the areal extent of major land covers within the adjoining watershed. The land uses in the Waquoit Bay watershed include areas of natural vegetation, freshwater ponds and wetlands, agricultural land, cranberry bogs, impervious surfaces (including roads, roofs, driveways), lawns and other turf, and houses (Fig. 2). Aerial photographs taken in 1938, 1944, 1951, 1955, 1971, 1980, 1985, and 1990 were incorporated into ArcInfo and analyzed to determine the areal cover represented by each type of land use (Brawley et al. 2000).

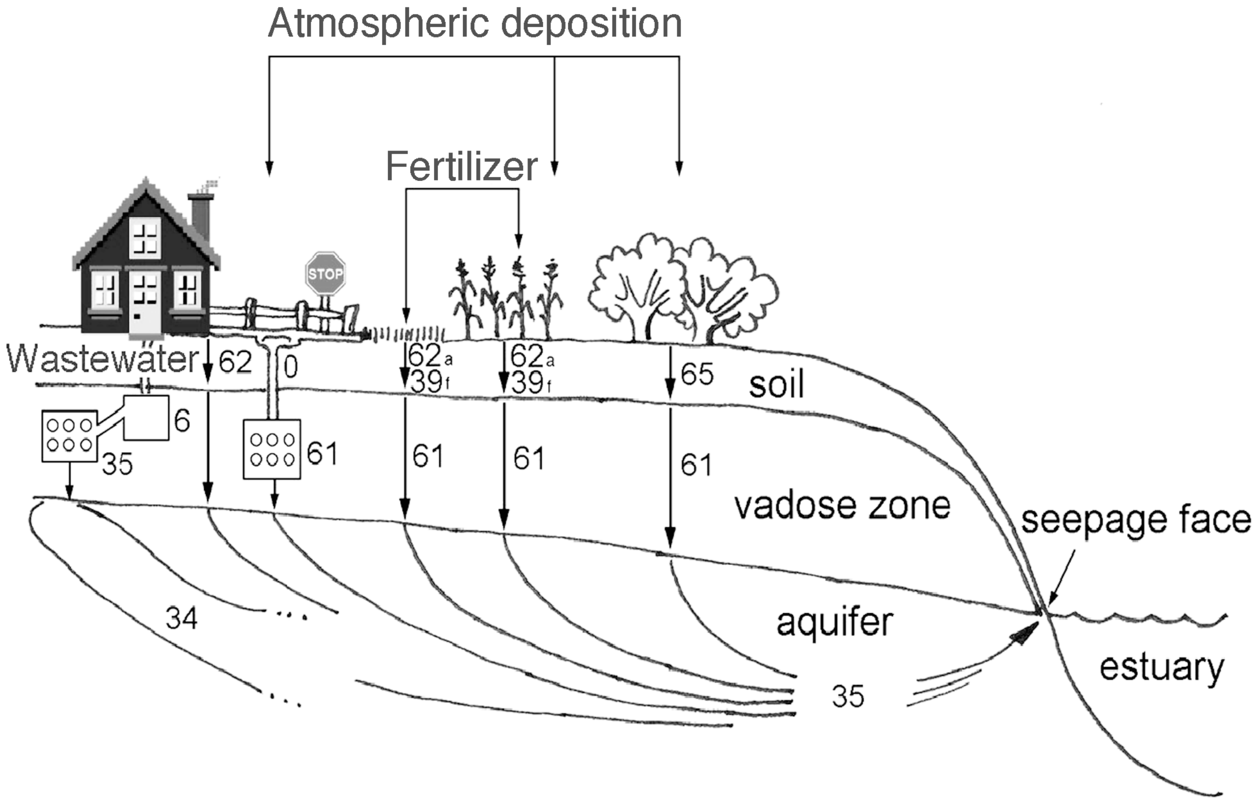
Changes in nitrogen sources

Atmospheric deposition

To estimate amounts of nitrogen deposited from the atmosphere during the period of study we used a historical reconstruction of nitrogen deposition to the northeastern region of the U.S.A. and the Maritime Provinces of Canada (Bowen and Valiela 2001). We compiled historical data on wet nitrogen deposition from 1910 to the present from extensive literature reviews. Historical data on the proportions of dry deposition and deposition of dissolved organic nitrogen (DON) do not exist in the literature, so we applied constant adjustment factors to the wet-deposition time course to estimate deposition of total nitrogen over the course of the century.

We estimated dry deposition and deposition of DON as follows. A dry deposition adjustment factor was developed using data from the National Dry Deposition Network and the Integrated Forest Studies Program. These data indicate that oxidized forms of dry nitrogen deposition were between 25 and 70% of oxidized wet deposition, and reduced forms were between 2 and 33% of wet NH_4-N deposition (Lovett and Lindberg 1993). In our calculation we used 48%, the mean of these ranges, as a dry-deposition factor for oxidized nitrogen and 18% as a dry-deposition factor for reduced forms of nitrogen (Bowen and Valiela 2001). Organic nitrogen ranged from 11 to 90% of total deposited nitrogen in different areas of the world (Timperley et al. 1985). We used an organic nitrogen adjustment factor of 52% of the total nitrogen deposition based on local measurements (Valiela et al. 1978) to account for

Fig. 2. Schematic diagram of the sources, land covers, and components of the watershed ecosystem that are included in the Waquoit Bay nitrogen-loading model (NLM). The numbers show losses of total nitrogen incurred as the nitrogen travels through the various components; *a* denotes the percentage of atmospheric nitrogen and *f* the percentage of fertilizer nitrogen lost as it passes through agriculture land covers. All numbers indicate percent loss of nitrogen entering each component (soil, vadose zone, aquifer, septic tank, leaching system, or storm drain). (From Valiela et al. 2000*a*, reproduced with permission of Biogeochemistry, Vol. © 2000 Kluwer Academic Publishers.)



deposition of organic nitrogen. Once we had established the reconstructed rate of nitrogen deposition, we applied it to the land-use mosaics of the Waquoit Bay watershed and used NLM to track the losses that occurred as the nitrogen traversed each land-use type.

Fertilizer use

Two pieces of information were required to estimate the nitrogen contribution from fertilizers. We needed to know the amount of fertilizer being applied to a given crop and the area of that crop within the watershed being modeled. Ideally, there would be historical records of fertilizer-application rates on all of the individual parcels within the watershed that was being modeled. However, that information is lacking, so as a reasonable alternative we estimated the fertilizer-application rate by using regional trends (USDA 1993, 1999) and applied that rate to the known land uses specific to our watershed.

We first obtained a regional estimate of historical changes in fertilizer-application rates using data for the northeastern U.S.A. (USDA 1993, 1999). We obtained estimates of total fertilizer consumption in New England and New York from 1900 to 1990, and of the total land area dedicated to major fertilized field crops (hay, corn, wheat, oats, and barley; USDA 1999). We then divided the total amount of fertilizer applied by the area under cultivation to roughly estimate fertilizer-application rates (kg·ha^{-1}) for these major field crops.

We applied the application rates developed for major crops to locally appropriate modern rates. On Cape Cod, fertilizers are more frequently used on smaller scale horticultural crops, lawns, golf courses, and cranberry bogs, but historical data comparable to those concerning fertilization of field crops were not readily available. In the absence of detailed data, we estimated the time course of fertilizer use for these other crops by using the historical trend line calculated for major field crops to backcast across the decades, starting from modern application rates. Cranberries are much less intensively fertilized than many other agricultural crops, so we used agricultural reports and bulletins produced throughout the century by the Massachusetts Cranberry Experiment Station to generate an independent time course for the use of fertilizers on cranberry bogs (Demoranville 1996).

Wastewater disposal

In some environments the distinction between wastewater nitrogen inputs and fertilizer nitrogen inputs is unclear. If a large portion of fertilized crops grown in a watershed is consumed in that watershed, then including both fertilizer and wastewater terms could overestimate the amount of nitrogen entering the system because the wastewater nitrogen would not be a new source of nitrogen but would be nitrogen transformed from the fertilizer inputs. In the Waquoit Bay watershed, as with many suburban landscapes, the vast majority of the foods that are consumed are imported from elsewhere. Thus, we feel justified in regarding wastewater nitrogen as a separate source of new nitrogen.

We assessed changes in wastewater-derived nitrogen according to the number of houses in the watershed, which was determined from aerial photographs taken in each of the years to be studied. Wastewater in the Waquoit Bay watershed is disposed of via on-site septic systems. We assumed a mean annual occupancy rate of 1.79 people per house (Valiela et al. 1997*a*). This occupancy rate reflects the seasonal nature of Cape Cod, where many residences are occupied only on weekends or in the summer (Valiela et al. 1997*a*). A compilation of literature values indicated that individuals release a mean of 4.8 kg N·year^{-1} (Valiela et al. 1997*a*). From the occupancy rate, number of houses, and amount of nitrogen produced per person each year, we calculated the total amount of wastewater-derived nitrogen released within the Waquoit Bay watershed.

Effect of historical changes in loads on eutrophication within the estuaries

To understand how changes in land use affect the ecology of coastal embayments, we used the time courses for each major source of nitrogen described above and entered the data into NLM to develop a historical trend in nitrogen loading into Waquoit Bay. To examine the ecological impact of the change in nitrogen loads we first used a space-for-time substitution to depict the changes that have occurred in the biomass of the three dominant primary producers, phytoplankton, macroalgae, and eelgrass (*Zostera marina*), in different estuaries of the Waquoit Bay estuarine complex that are currently exposed to different nitrogen loads. We used previous studies on phytoplankton and macroalgal biomass in the different sub-estuaries of Waquoit Bay for the space-for-time substitution (Valiela et al. 2000*b*). We then used our historical reconstruction of loads from the Waquoit Bay watershed, plus aerial photographic evidence (Costa 1988) and visual surveys (Short and Burdick 1996) of seagrass cover, to assess how urbanization may have affected the ecology of the Waquoit Bay estuarine system. Finally, we traced the changes that have occurred in the estuary to the level of animals by looking at historical changes in the scallop harvest as a proxy for changes in the food webs of receiving estuaries.

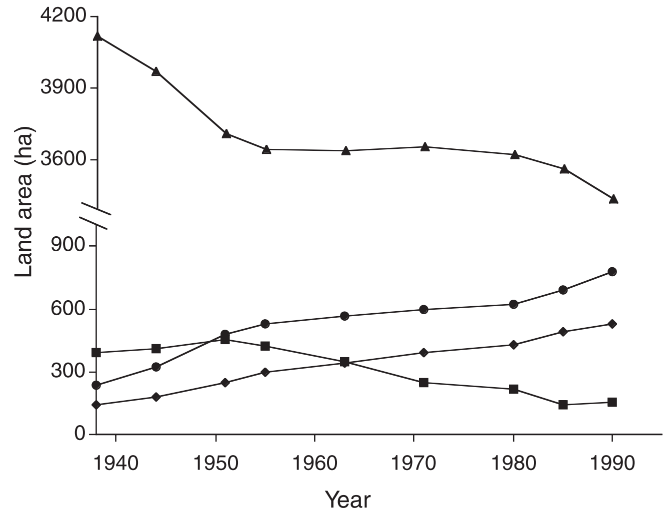
Results and discussion

Substantial changes have occurred in the Waquoit Bay watershed during the last several decades as the watershed became more urbanized. Below we examine the changes in the land-use mosaic, as well as how the sources of nitrogen have changed, how that is reflected in the nitrogen load entering Waquoit Bay, and how the ecology of the Bay has changed as a result of the alteration of land use in its watershed during the last 60 years.

Changes in the land-use mosaic

Land use in the Waquoit Bay watershed changed substantially between 1938 and 1990 (Fig. 3). The areas of fertilized turf (golf courses, lawns, and other turf areas) and impervious surfaces (roads, roofs, driveways, and parking lots) within the watershed have all increased since 1938. These increases are largely associated with urbanization. Lawns accounted for 33% of the increase in turf area between 1938 and 1990. An additional 58% of the increase in turf area is classified as "other," but this is predominantly associated with an increase in turf area surrounding the Massachusetts Military Reservation at the northern extreme of the watershed. The golf courses in the Waquoit Bay watershed account for only 6% of the total turf area and less than 1% of the total land area. Roads, roofs, and driveways account for 44% of the increase in area of impervious surfaces. The remaining increases in impervious-land area result mostly from increases in the number of parking lots associated with commercial developments within the watershed, as well as commercial areas and runways in the Massachusetts Military Reservation.

Fig. 3. Changes in land use within the Waquoit Bay watershed from 1938 to 1990. Turf area (●) includes lawns, golf courses, and parks. Impervious surfaces (◆) include roads, roofs, driveways, and parking lots. Agricultural land (■) includes horticultural areas and cranberry bogs. Natural vegetation (▲) is calculated as the total land area minus all non-natural land covers.



Pressure for residential use of land parcels has caused losses of naturally vegetated areas, cranberry bogs, and other agricultural areas since the 1930s (Fig. 3). Naturally vegetated lands decreased from 84% of the total land area in the Waquoit Bay watershed in 1938 to only 68% in 1990. Agricultural areas and cranberry bogs also decreased from a peak of 553 ha in 1955 to only 225 ha in 1990.

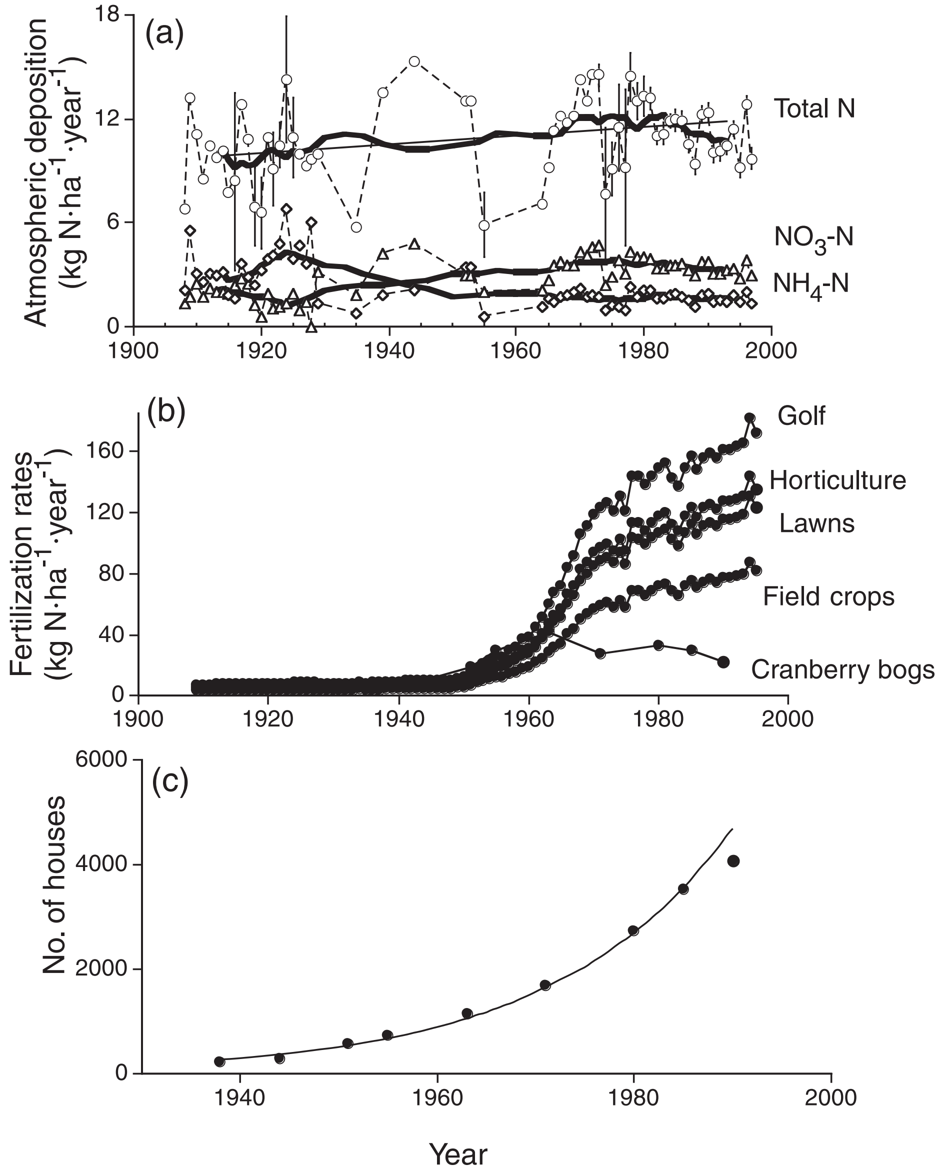
Changes in nitrogen sources

Atmospheric deposition

Wet deposition of ammonium decreased significantly (F = 34.6, P < 0.01; Bowen and Valiela 2001) through the century, while wet deposition of nitrate increased significantly (F = 53.9, P < 0.01; Bowen and Valiela 2001; Fig. 4*a*). The shift in dominance between ammonium and nitrate wet deposition may be another consequence of increased urbanization in the northeastern U.S.A., since anthropogenic ammonium deposition is largely due to emissions from livestock waste and fertilized soils (Dentener and Crutzen 1994), both of which have decreased regionally over the last half of the 20th century. The largest anthropogenic source of NO_x emissions is the combustion of fossil fuels during industrial and commercial processes and in gasoline engines. Nitrogen from all of these sources has increased as a result of increased urbanization on Cape Cod (Bowen and Valiela 2001), and the increased industrialization of the Midwest, whose NO_x emissions contribute substantially to atmospheric deposition in the northeastern U.S.A. (Dennis 1995).

Total nitrogen deposition to the Massachusetts area increased significantly ($y = 0.026x - 39.4$, F = 41.2, P < 0.01) during the 20th century, by a rate of around 0.26 kg N·ha^{-1} per decade (Fig. 4*a*), and is presently approximately 12 kg N·ha^{-1}· year^{-1}. The vertical lines in Fig. 4*a* show the stan-

Fig 4. (*a*) Ammonium (NH_4-N), nitrate (NO_3-N), and total nitrogen deposition onto the Cape Cod area from 1910 to 1995. Data for ammonium (◇) and nitrate (△) represent wet deposition only; total nitrogen (○) includes estimates of dry deposition and deposition of organic nitrogen. The symbols represent the year-to-year data and the thick line is the 10-year moving average of the data. Error bars are included for total nitrogen when data are from more than one study (data from Bowen and Valiela 2001). (*b*) Reconstructed trends in fertilizer-application rates on golf courses, horticultural crops, lawns, field crops, and cranberry bogs in the northeastern U.S.A. between 1910 and 1995. (*c*) Numbers of houses in the Waquoit Bay watershed between 1938 and 1990 ($Y = 0.0 \times 10^{0.02x}$, $R^2 = 0.99$).



dard deviation of the means for total nitrogen in years for which more than one value was reported in the literature. The mean standard deviation for any one point estimate of total deposition is 3.24 kg N·ha^{-1}·year^{-1}. Our interest was to define a long-term trend; we can assess the uncertainty in the slope of the regression fitted to the entire data set by the ratio of the standard error of the slope, 0.009, relative to the slope of the regression, 0.026. This coefficient of variation was 34% of the slope. This represents considerable uncertainty, but does not prevent the regression from being significant.

There is some ambiguity in interpreting these loads because there is a question about the lability of the organic component of atmospheric nitrogen. The conventional wisdom has always been that organic nitrogen is largely refractory and so its role in ecological processes must be modest. Seitzinger and Sanders (1999) suggest that 45–75% of atmospheric organic nitrogen could be labile. In Waquoit Bay we find that forested watersheds contribute groundwater that is high in organic nitrogen (Valiela et al. 2000*b*), but the receiving estuaries are not eutrophic. This suggests that atmospherically derived organic nitrogen might be less biologically available than organic nitrogen from urbanized watersheds. The lability of atmospheric DON thus still needs study.

The magnitude of the nitrogen load from the atmosphere is of some significance, because at certain rates of deposition the ability of the forest to intercept atmospheric nitrogen declines, a phenomenon described as nitrogen saturation (Aber et al. 1989; Nadelhoffer et al. 1995). Evidence of nitrogen saturation was visible in some Welsh forests when total nitrogen inputs exceeded 10 kg·ha^{-1}·year^{-1} (Emmett et al. 1993). When total nitrogen deposition exceeded 25 kg·ha^{-1}·year^{-1} all sites studied displayed nitrogen leaching (Dise and Wright 1995). Our reconstructed record of nitrogen deposition indicates that Cape Cod forests are receiving inputs of atmospheric nitrogen that are high enough to induce nitrogen saturation. If there are additional increases in the deposition of total atmospheric nitrogen, this will inevitably result in further leaching of nitrogen to coastal waters.

Fertilizer use

From 1913, when the first commercial ammonium factory opened, until the 1940s, less than 4.5 t of fertilizers were synthesized (Smil 1997). Today roughly 80 million t are being produced globally (Smil 1999). The use of nitrogen fertilizers in the northeastern U.S.A. increased markedly during the same time period. The mean dosage of fertilizer applied to field crops in the northeastern U.S.A. was relatively constant between 1910 and 1940, but has since risen sharply, increasing from less than 10 kg N·ha^{-1}·year^{-1} in 1950 to 80 kg N·ha^{-1}·year^{-1} in recent years (Fig. 4*b*). The rate of increase in fertilizer production appears to be slowing, both in New England (Fig. 4*b*) and globally (Frink et al. 1999).

Fertilizer-application rates to non-agricultural lands can be even higher than application rates to some crops. We estimated secular trends in fertilizer-application rates for most of the dominant uses of fertilizer on Cape Cod. Fertilizer application to turf, particularly golf courses, involves the largest dosages of nitrogen (161 kg·ha^{-1}·year^{-1}; Correll et al. 1992), followed by applications of nitrogen to horticultural crops (136 kg·ha^{-1}·year^{-1}; Petrovic 1990) and lawns (122 kg·ha^{-1}·year^{-1}; Valiela et al. 1997*a*). Today only 20–25 kg N·ha^{-1}·year^{-1} is applied to cranberry bogs, a decrease from a peak of about 45 kg N·ha^{-1}·year^{-1} in the 1960s (Demoranville 1996).

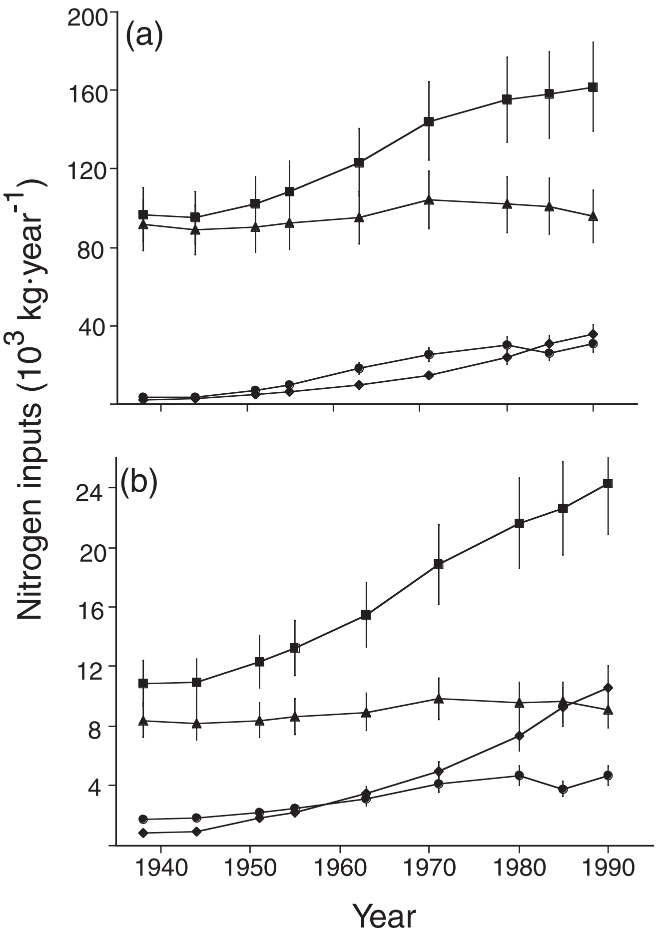
Wastewater disposal

Pressure for residential use of land can be seen in the rise in the number of houses in the Waquoit Bay watershed during the last 60 years (Fig. 4*c*), from fewer than 250 to more than 4000. The total amount of wastewater generated within the Waquoit Bay watershed increased from just over 2000 kg N·year^{-1} in 1938 to almost 30 000 kg N·year^{-1} in 1990, a factor of almost 17. This sharp increase demonstrates the impact of urban sprawl on these coastal watersheds.

Nitrogen loading into Waquoit Bay

Total nitrogen inputs into the Waquoit Bay watershed increased almost twofold between 1938 and 1990 (Fig. 5*a*). Atmospheric deposition has been the predominant contributor of nitrogen to the watershed during the last 60 years, but the magnitude of this deposition onto the Waquoit Bay landscape has remained surprisingly uniform throughout the century. The relative importance of atmospheric deposition has decreased: in 1938 atmospheric deposition onto the watershed accounted for 95% of the nitrogen input, but in 1990 it accounted for only 59% of total nitrogen delivery. Wastewater and fertilizer were minor sources of nitrogen in midcentury, but increased to 22 and 19% of the total contribution, respectively (Fig. 5*a*, Table 1). It was clearly the increase in wastewater and fertilizer nitrogen that was responsible for the increase in nitrogen load between 1938 and 1990.

Fig. 5. Modeled historical nitrogen loads into the watershed (*a*) and estuary (*b*) of Waquoit Bay. The total nitrogen loads (■) are broken down according to the three major sources of nitrogen: atmospheric deposition (▲), wastewater disposal (◆), and fertilizer application (●). Data are plotted with the 14% standard error that is associated with the Waquoit Bay NLM (Valiela et al. 1997*a*).



After losses and transformations that occur during transport through the landscape and aquifer are accounted for, NLM calculations showed that delivery of nitrogen to the estuary more than doubled from 1938 to 1990 (Fig. 5*b*, Table 1). Atmospheric deposition accounted for 77% of the total nitrogen delivered to Waquoit Bay in 1938 and only 37% of the load in 1990. Wastewater derived from on-site septic systems increased by a factor of more than 12, and in 1990 accounted for 43% of all of the nitrogen entering Waquoit Bay (Table 1). Wastewater, because it is subject to lower losses within the watershed, became the most important source of nitrogen entering the estuary during the 1980s, despite the fact that atmospheric nitrogen was the largest source of nitrogen entering the watershed.

The relative contributions by wastewater disposal, fertilizer use, and atmospheric deposition to nitrogen loading into

Table 1. Relative contributions of the major sources of nitrogen entering the Waquoit Bay estuary in 1938 and 1990.

	1938 nitrogen load		1990 nitrogen load	
Source of nitrogen	$\times 10^3$ kg·year^{-1}	%	$\times 10^3$ kg·year^{-1}	%
Entering the watershed				
Atmospheric deposition	91.3	95	95.5	59
Wastewater disposal	2.1	2	35.7	22
Fertilizer use	3.2	3	30.5	19
Total	96.6	100	161.7	100
Entering the estuary				
Atmospheric deposition	8.4	77	9.1	38
Wastewater disposal	0.7	7	10.5	43
Fertilizer use	1.7	16	4.7	19
Total	10.9	100	24.3	100

Note: The propagated standard error of the modeled nitrogen load is 14% (Valiela et al. 1997*a*). The percent contribution from each nitrogen source is slightly different from values published in Valiela et al. (1997*a*). The differences result from the need to use regional trends in order to incorporate historical changes. In some instances these regional trends were slightly different from the specific information on Waquoit Bay used in the original publication. The difference between the regional approach and Valiela et al.'s (1997*a*) results fall within the standard error of the model.

Table 2. Percent contributions of the major sources of nitrogen to various Massachusetts and Rhode Island coastal ponds.

	Source of nitrogen (%)			Population density (no. of people·ha^{-1})
Estuary	Wastewater	Fertilizer	Atmosphere	
Sage Lot Pond, Mass.	5	46	49	0.21
Trustom Pond, R.I.	17	70	13	0.75
Cards Pond, R.I.	21	68	11	0.96
Waquoit Bay, Mass.	43	20	37	1.45
Mashpee River, Mass.	49	15	36	1.81
Green Pond, Mass.	54	25	21	1.81
Pt. Judith Pond, R.I.	55	36	9	2.14
West Falmouth Harbor, Mass.	64	6	30	2.47
Childs River, Mass.	60	12	28	2.54
Green Pond, R.I.	47	43	10	3.65
Potter's Pond, R.I.	50	39	11	4.62
Coastal ponds (mean ± SD)	42 ± 19	35 ± 21	23 ± 13	2.03 ± 1.3
Contribution from northeastern U.S.A. to North Atlantic Ocean[a]	33 (26)	22 (17)	44 (34)	0.87

Note: Data for Rhode Island Pond are from Lee and Olsen (1985) and Nixon et al. (1982) and are based on 1981 land use. Data for Massachusetts are modeled using the Waquoit Bay nitrogen-loading model (NLM) and are based on 1990 land use. Data for Green Pond are from Kroeger et al. (1999). Data for the northeastern U.S.A. are from Howarth et al. (1996).

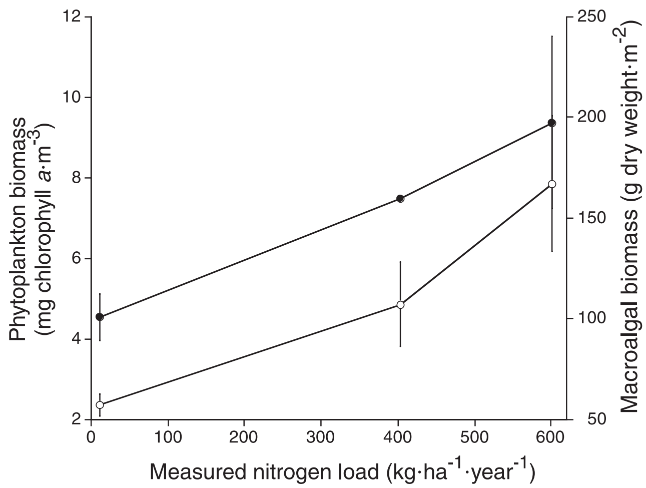
[a]Howarth et al. (1996) include biological nitrogen fixation as a source that contributes 23% of the nitrogen to the North Atlantic Ocean. Numbers in parentheses show the percent contribution from biological nitrogen fixation; non-parenthetical numbers do not include biological fixation.

estuaries throughout the northeastern U.S.A. vary (Table 2). Nitrogen contributions from wastewater range from 5 to 64% of the total nitrogen load, those from fertilizer range from 6 to 70%, and those from atmospheric deposition range from 9 to 49%. There is variation from one estuary to another, but the specific value depends on the size of and relative land uses in the watershed. In some cases, inputs of agricultural fertilizers are dominant. Chesapeake Bay, for example, receives 36% of its nitrogen from fertilizers (Hinga et al. 1991), and large portions of the Chesapeake Bay watershed are under agricultural land use. Elsewhere, urban sprawl drives the increased nitrogen inputs. In Massachusetts and Rhode Island estuaries the percent contributions of wastewater disposal and population density (Table 2) generally increased together. In general, although there is some variation from one watershed to another, the mean contribution of nitrogen from wastewater disposal, atmospheric deposition, and fertilizer use to these estuaries is statistically similar to the calculated loads from the entire northeastern U.S.A. (Howarth et al. 1996).

Impact on producer communities and estuarine food webs

The link between land uses, nitrogen loads, and estuarine

Fig. 6. Biomass of phytoplankton (●) and macroalgae (○) in estuaries with increasing nitrogen loads. From left to right the estuaries are Sage Lot Pond, Quashnet River, and Childs River. Data are taken from Valiela et al. (2000*b*).



food webs has been established using stable-isotopic evidence in estuaries receiving different nitrogen loads. This linkage occurs because wastewater-derived nitrogen has a higher ^{15}N isotopic signature than nitrogen derived from either atmospheric or fertilizer sources (Gormly and Spaulding 1979; Kreitler 1979). As the proportion of the watershed that is urbanized increases, so does the isotopic signal in the groundwater and in the water column (McClelland and Valiela 1998). When that nitrogen is incorporated by estuarine producers, the heavy ^{15}N signal is incorporated as well, so producers in estuaries receiving a greater proportion of wastewater-derived nitrogen have heavier isotopic signals than producers in pristine estuaries (McClelland and Valiela 1998). The ^{15}N signal can even be traced through both primary and secondary consumers (McClelland and Valiela 1998). Thus, it is possible to directly follow the nitrogen derived from wastewater through the entire food web in estuaries. The incorporation of wastewater-derived nitrogen into the producer and consumer populations, however, does not necessarily indicate that there have been alterations in the structure of estuarine food webs. Were the changes in land use sufficient to alter the ecology of the receiving estuaries?

The biomass of phytoplankton and benthic macroalgae in Waquoit Bay increased across a range of estuaries receiving increasingly higher nitrogen loads (Fig. 6). These direct responses to nitrogen enrichment (Valiela et al. 1997*b*) have several indirect repercussions for the rest of the ecosystem. Phytoplankton and macroalgal mats derive their nutrients largely from the water column. Seagrasses, however, tend to be light, rather than nutrient, limited (Duarte 1995). When nitrogen is added to estuaries, because of their increased growth, phytoplankton and macroalgae intercept increasing portions of light, thereby limiting light availability for seagrasses (Hauxwell et al. 2001).

Growth, areal cover, and biomass of eelgrass diminished in response to increases in phytoplankton and macroalgal biomass (Hauxwell et al. 2001). Areal cover of eelgrass, when viewed across a range of northeastern U.S. estuaries, was sharply reduced at nitrogen loads greater than 20 $kg \cdot ha^{-1} \cdot year^{-1}$; the meadows disappeared by the time nitrogen loads exceeded 100 $kg \cdot ha^{-1} \cdot year^{-1}$ (Fig. 7*a*). Temporal changes that have occurred during the last 60 years in Waquoit Bay also demonstrate a sharp decline in eelgrass area (Fig. 7*b*). Before the 1950s, eelgrass was still recovering from the near-complete loss caused by the wasting disease of the early 1930s (Renn 1935). Note also that during the 1930–1950 period, nitrogen loads were lower than the 20 $kg \cdot ha^{-1} \cdot year^{-1}$ that we posit as a threshold for the survival of eelgrass meadows. The building boom on Cape Cod during the 1960s resulted in an increase in nitrogen loads, so that by the early 1970s, the nitrogen load exceeded 20 $kg \cdot ha^{-1} \cdot year^{-1}$, eelgrass meadows were notably smaller in area, and the loss of eelgrass habitat continued through 1990. The historical reconstruction indicates that the nitrogen loads corresponding to near complete destruction of eelgrass meadows ranged only between 15 kg and 30 $kg \cdot ha^{-1} \cdot year^{-1}$ (Fig. 7*b*).

Since the presence of seagrass is required for the maintenance of many taxa, including commercially important shell- and fin-fish species, changes in eelgrass cover imply drastic changes in the rest of the food webs in affected estuaries. For example, the bay scallop (*Argopecten irradians*), a commercially important shellfish species, is closely dependent on eelgrass habitat. During the time-span when the eelgrass meadow area decreased (Fig. 7*b*), the annual harvest of bay scallops in Waquoit Bay decreased by three orders of magnitude, from over 100 000 $L \cdot year^{-1}$ in the early 1960s to under 100 $L \cdot year^{-1}$ in recent years (Fig. 8*a*). The decline in the annual scallop harvest correlated with an increase in nitrogen loads (Fig. 8*b*). Ryther and Dunstan (1971) reported a similar loss of oysters and hard clams as a result of nutrient enrichment from the Long Island duckling industry in Moriches Bay, N.Y. We can therefore claim that changes in land use in watersheds, to urban sprawl in particular, can be demonstrably linked to a drastic restructuring of estuarine ecosystems.

Implications for the management of coastal watersheds

The historical reconstruction of land-derived nitrogen loads, plus the linking of these loads to sensitive indicators such as eelgrass meadows and bay scallops, provides some means to identify management priorities as well as define management end-points. If management goals include a reduction of nitrogen loads, the results of this work suggest some general approaches which could be used to remediate the increased nitrogen loads that are the result of urban sprawl in coastal watersheds.

Preservation of green open space on land can make a contribution to the management of nitrogen loads. Atmospheric-nitrogen deposition is still the largest nitrogen source in the watershed of Waquoit Bay. Interception of atmospheric nitrogen is highest where natural vegetation covers land parcels, and this feature needs to be a part of the tool kit of a manager thinking about the coastal zone. Preservation of green space may slow the increase in nutrient enrichment to receiving waters by intercepting atmospheric deposition, and also by limiting the supply of nitrogen that would result from conversion to other land uses like agriculture or residential building.

Management of wastewater and restriction of the use of fertilizers are the most practical measures available locally to lower nutrient enrichment. Zoning restrictions, improve-

Fig. 7. (*a*) Percentage of eelgrass area lost as a function of nitrogen load. The data, for various Massachusetts estuaries, are taken from Valiela et al. (2001). (*b*) Percentage of area covered with eelgrass in Waquoit Bay estimated from aerial photographs taken between 1940 and 1989 (Costa 1988; Short and Burdick 1996) plotted against modeled historical nitrogen loads.

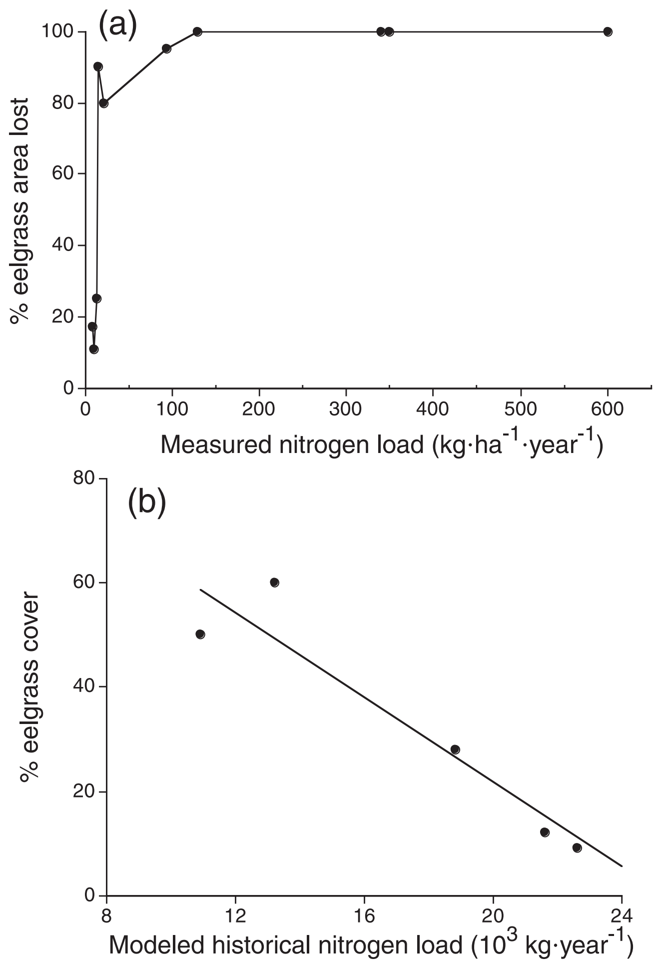
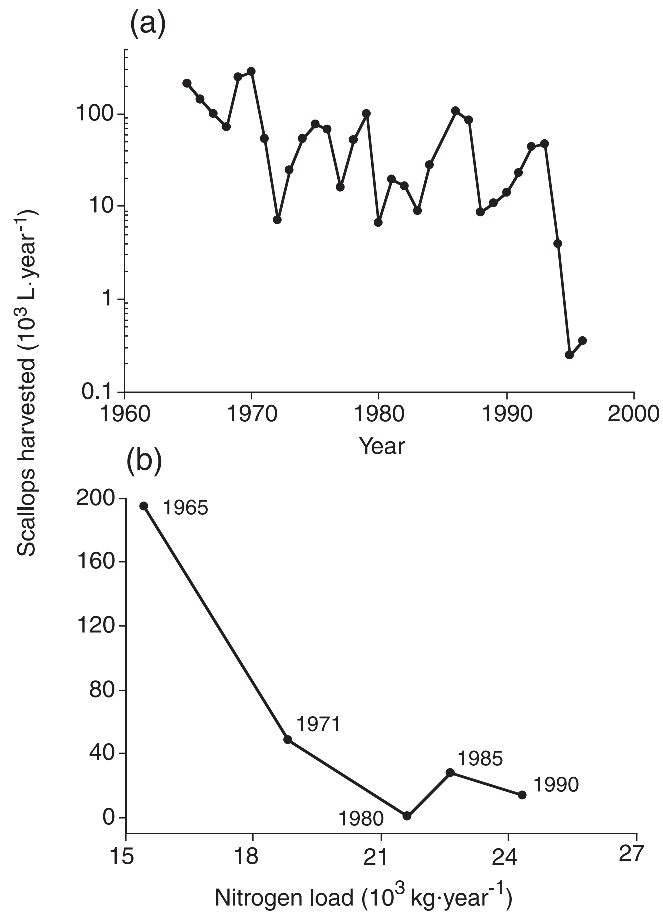


Fig. 8. (*a*) Amounts of scallops reported to have been harvested in Waquoit Bay from 1960 to 1997. (*b*) Volumes of scallops reported to have been harvested in Waquoit Bay as a function of the modeled historical nitrogen loads from this reconstruction. Data are from the Shellfish Warden's annual report included in the Town of Falmouth Annual Reports.



ment of in situ septic systems, or installation of small sewage-treatment plants could all lower the inputs of wastewater nitrogen. Restricting the use of fertilizers and area of turf will also, to a small degree, lower nitrogen inputs, but could be especially effective in areas with more intensive agriculture. Choice of the appropriate management option depends on the target a community wants to achieve. Stakeholders might decide, for example, that they would like to restore eelgrass to 50% of their estuary. Then, modeling such as we have done in this paper could identify the nitrogen load that was present at the time when such eelgrass cover existed, and indicate the pathways that would possibly reduce nitrogen loads.

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